

Ground Water Quality

Reduction of Nitrate Leaching with Haying or Grazing and Omission of Nitrogen Fertilizer

L. B. Owens* and J. V. Bonta

ABSTRACT

In some high-fertility, high-stocking-density grazing systems, nitrate (NO_3) leaching can be great, and ground water $\text{NO}_3\text{-N}$ concentrations can exceed maximum contaminant levels. To reduce high N leaching losses and concentrations, alternative management practices need to be used. At the North Appalachian Experimental Watershed near Coshocton, OH, two management practices were studied with regard to reducing $\text{NO}_3\text{-N}$ concentrations in ground water. This was following a fertilized, rotational grazing management practice from which ground water $\text{NO}_3\text{-N}$ concentrations exceeded maximum contaminant levels. Using four small watersheds (each approximately 1 ha), rotational grazing of a grass forage without N fertilizer being applied and unfertilized grass forage removed as hay were used as alternative management practices to the previous fertilized pastures. Ground water was sampled at spring developments, which drained the watershed areas, over a 7-yr period. Peak ground water $\text{NO}_3\text{-N}$ concentrations before the 7-yr study period ranged from 13 to 25.5 mg L^{-1} . Ground water $\text{NO}_3\text{-N}$ concentrations progressively decreased under each watershed and both management practices. Following five years of the alternative management practices, ground water $\text{NO}_3\text{-N}$ concentrations ranged from 2.1 to 3.9 mg L^{-1} . Both grazing and haying, without N fertilizer being applied to the forage, were similarly effective in reducing the $\text{NO}_3\text{-N}$ levels in ground water. This research shows two management practices that can be effective in reducing high $\text{NO}_3\text{-N}$ concentrations resulting from high-fertility, high-stocking-density grazing systems, including an option to continue grazing.

IN AN INCREASINGLY larger number of research papers, there are reports of pasture systems and situations that cause detrimental effects on water quality. Most of these grassland studies have focused on nitrate (NO_3) leaching. Research in New Zealand and England has shown NO_3 subsurface losses from grasslands to be much greater when grazed, especially by cattle (Sharpley and Syers, 1979; Ryden et al., 1984; Steele et al., 1984; Haigh and White, 1986). Ridley et al. (1999) also noted that pastures receiving high N treatments have the potential to contaminate streams and ground water. Research in Ohio has shown NO_3 leaching to be greater from pastures receiving high N fertilization (Owens et al., 1983, 1992) than from moderate N fertilization (Owens et al., 1982) or under a grass-legume mixture (Owens et al., 1994). In England, Tyson et al. (1997) reported that N leaching losses from grass-white clover (*Trifolium repens* L.) pastures were only 26% of the amount re-

corded from fertilized grass pastures, although annual liveweight gain of the grazing steers was lower also.

The concern for NO_3 leaching has increased as some grazing practices have shifted to much greater animal density in the grazing systems. Such systems, where forage and animal production are maximized, create more deposition of urine and feces (i.e., spots with high concentrations of N). There have been lysimeter studies comparing N leaching with and without urine applications (Silva et al., 1999; Stout et al., 2000; Di and Cameron, 2002), and the results showed NO_3 leaching in excess of water quality standards with urine applications, especially in connection with high N fertilizer applications.

Some research has been done with management practices to reduce or prevent high N concentrations in water leaching from pastures. Di and Cameron (2002) noted that large amounts of $\text{NO}_3\text{-N}$ could be lost via leaching but this could be diluted in aquifers that receive recharge from ungrazed areas. In a German study, Anger et al. (2002) compared NO_3 leaching from intensive grazing (4.9 livestock units ha^{-1} with 250 kg N ha^{-1} applied annually) with extensive grazing (2.9 livestock units ha^{-1} with no N fertilizer on a pasture comprised of up to 15% legumes by dry matter). They found that NO_3 leaching could be reduced with the extensive grazing management nearly to levels found under mown grassland. A New Zealand study (de Klein and Ledgard, 2001) compared a "conventional grazing system" with a "restricted grazing system" (grazing is restricted to times when the risk of NO_3 leaching is smallest) and a "nil grazing system" (forage is harvested and fed to the cows; excreta is collected and returned to the field). Nitrate leaching losses were reduced by 35 to 50% and 55 to 65% for the restricted and nil systems, respectively, compared with the conventional grazing system.

Although many of the environmental concerns associated with high N fertilization of pastures and/or intensive grazing relate to leaching pathways, there are other major pathways of N loss. There can be considerable gaseous N losses from grasslands via the processes of N_2O emissions, denitrification, and NH_3 volatilization. Greater amounts of N are lost via each pathway with greater N inputs (Fowler et al., 1997; Oenema et al., 1998; Ryden, 1986; Whitehead, 1995, p. 191–192).

Even with multiple pathways of N loss from grazing systems, there are management situations in which N concentrations and losses moving through the soil into

USDA-ARS, P.O. Box 488, Coshocton, OH 43812. Received 3 Mar. 2003. *Corresponding author (owens@coshocton.ars.usda.gov).

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677 S. Segoe Rd., Madison, WI 53711 USA

Abbreviations: WS, watershed.

ground water create adverse environmental effects. Changes in management practices can reduce the amount of NO₃ leaching and lower NO₃-N concentrations in subsurface flow to acceptable water quality levels. The purpose of this research was to evaluate reduction of NO₃ leaching and ground water N concentrations with two different, nonfertilized grassland management practices using small watershed systems.

MATERIALS AND METHODS

The study was conducted at the North Appalachian Experimental Watershed near Coshocton, Ohio. A 17.2-ha study area was divided into four paddocks (Fig. 1), each with an instrumented watershed for determining surface runoff. Precipitation and subsurface water-flow measurement sites and a battery of four lysimeters (Harrold and Dreibelbis, 1958) were also located in the study area.

Station precipitation is continental and conforms to the Ohio River Valley pattern. The watershed slopes range from 12 to 25% with an average of 20%. Soils are well-drained residual silt loams (Typic Dystrochrepts and Hapludults). The subsurface measurement sites (Fig. 1) were located at springs on the outcrop of the Middle Kittanning clay, a nearly impermeable layer that maintains a perched aquifer in the study area. The soils, climate, geology, and geomorphology of the area were described by Kelley et al. (1975).

During the 11 years before the current study (1979–1990), the vegetation was predominantly orchardgrass (*Dactylis glomerata* L.) and Kentucky bluegrass (*Poa pratensis* L.). A spring calving herd of beef cows (*Bos taurus*) grazed the pastures rotationally during the summer (May–October) and were fed hay, grown elsewhere on the station, in one pasture (Watershed [WS] 129) during the dormant winter period (November–April) until 1986. Then cattle were no longer wintered in this study area. Beginning in April 1979, the annual fertilizer N rate for the summer-grazed-only pastures was 168 kg ha⁻¹ and split equally among three applications. Methylene urea, a slow-release N fertilizer (39–0–0, containing 37.5% water-insoluble N), was applied to the pastures containing WS 102 and 104; NH₄NO₃ (34–0–0, no water-insoluble N) was applied to WS 135 and its surrounding pasture (Fig. 1). Because of the high rate of N brought into the winter feeding paddock with hay (297 kg ha⁻¹ annually; Owens et al., 1982), no N fertilizer was applied during this study period. Management details were described by Owens et al. (1992).

From May 1990 through April 1997, the cattle herd was rotated through two paddocks (WS 102 and 135) with the same frequency as with previous management. Hay was harvested and removed from the paddocks containing WS 104 and 129. When the cattle would normally have grazed these paddocks, they were moved to grazing areas outside of the study area. No N fertilizer was applied to any of these paddocks during this study period. Although there was merit in having a positive control (i.e., one watershed continued with the 168 kg N ha⁻¹ rate), we considered it more important to replicate each of the management practices studied for reduction of N leaching. It was clear that the 168 kg N ha⁻¹ management was not environmentally sound, and alternatives needed to be studied. Because of the winter-feeding on WS 129, NO₃-N leaching was greater than from the other watersheds. Even though it had the highest peak NO₃-N concentrations in subsurface flow, it offered an excellent opportunity to study reduction in NO₃-N leaching by a nonfertilized forage management practice. Achieving similarly low NO₃-N concen-

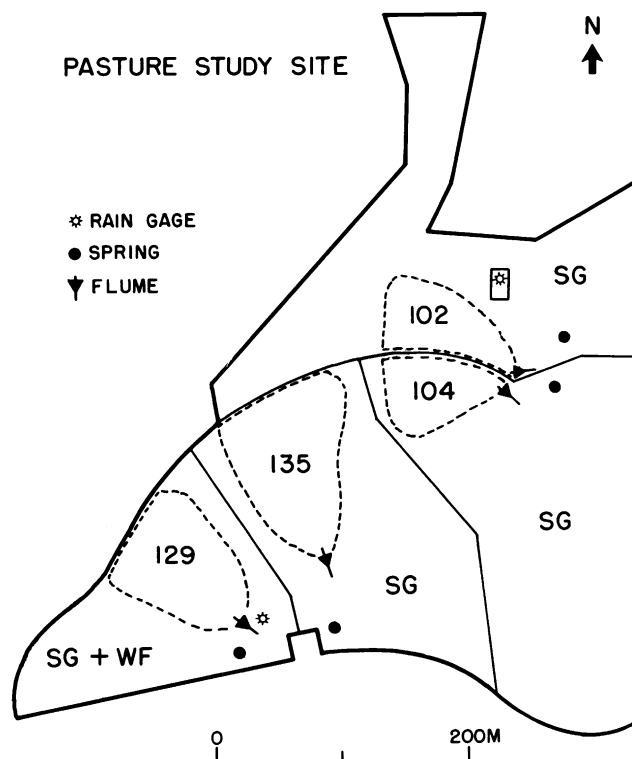


Fig. 1. Map of the 17.2-ha study area, showing four pastures (solid lines), gaged watersheds (dashed lines), rain gages, and spring-flow measurement sites. Watersheds (WS) 102 and 135 were grazed rotationally; WS 104 and 129 had forage removed as hay.

trations from different starting points should confirm the NO₃-N reduction effectiveness of the management practice.

Surface runoff from the watersheds was measured by pre-calibrated 0.76-m H-flumes. Surface runoff water samples were collected during each runoff event by Coshocton wheels (Brakensiek et al., 1979) modified to continuously deliver a flow-proportional sample of runoff water to a refrigerated container. Precipitation was measured by two recording rain gages located in the study area. Because rain gage measurements have been shown by weighing lysimeters to underestimate true catch at ground level, gage readings were corrected (McGuinness, 1966) to estimate "true" precipitation input.

Subsurface flow was continuously measured with HS and V-notch flumes and sampled monthly by hand. The subsurface basin is larger than the surface watershed, and the quantity of subsurface flow contributed from beneath a watershed is calculated with the aid of lysimeter data (Van Keuren et al., 1979). Using data from 1984 through 1999, the calculated subsurface flow for the individual watersheds correlated strongly with the measured subsurface flow from the subsurface basin (r^2 ranged from 0.6 to 0.8).

Water samples (surface runoff, subsurface flow, and rainfall) were immediately filtered after collection and refrigerated at 4°C until analyzed. Analyses were performed within a few days of sample collection. Analyses for NH₄-N were conducted using an automated phenate method, and for NO₃-N plus NO₂-N by an automated Cd-reduction method (USEPA, 1979). Total N was determined by an automated phenate method after digestion in a block digester (Schuman et al., 1973) modified to include NO₃-N and NO₂-N. Organic N was obtained by difference between total N and mineral N (the sum of NH₄-, NO₃-, and NO₂-N). Nitrogen content of hay was determined by the Kjeldahl method (Association of Official Analytical Chemists, 1984).

Table 1. Average seasonal and annual N concentrations and inputs in precipitation for the study period and the seven years before the study period.

		Concentration		Total input	
Time	Precipitation	NO ₃ -N	Mineral N†	NO ₃ -N	Mineral N†
	mm	mg L ⁻¹		kg ha ⁻¹	
Seven years before study period (1983–1990)					
Growing season‡	530 ± 81§	0.9 ± 0.4	1.9 ± 1.2	4.2 ± 1.9	8.0 ± 3.2
Dormant season¶	473 ± 86	0.9 ± 0.8	1.7 ± 1.0	3.9 ± 4.3	7.0 ± 4.8
Year#	1003 ± 63	0.9 ± 0.6	1.8 ± 1.1	8.1 ± 6.6	15.0 ± 3.6
Seven-year study period (1990–1997)					
Growing season	542 ± 161	1.0 ± 0.4	2.2 ± 1.1	4.5 ± 1.9	9.8 ± 5.0
Dormant season	527 ± 76	1.0 ± 0.5	1.7 ± 0.8	4.6 ± 2.0	8.0 ± 3.5
Year	1069 ± 217	1.0 ± 0.4	1.9 ± 1.0	9.1 ± 3.9	17.8 ± 8.9

† Sum of NH₄-N, NO₃-N, and NO₂-N.

‡ May–October.

§ Values are means and standard deviations, calculated for the seasons using monthly data within those seasons.

¶ November–April.

May–April.

The chemical data were tabulated on a seasonal basis corresponding to the cattle-management treatments, that is, a 6-mo (May–October) growing–grazing period and a 6-mo (November–April) dormant period. The transport of each chemical for a season is the sum of the transport (i.e., flow times concentration) for all samplings within the season. The seasonal concentration values are flow-weighted averages calculated by dividing the total seasonal chemical transport by the total water flow for that season.

RESULTS AND DISCUSSION

Precipitation

Before considering the N losses from the watersheds, N inputs should be assessed. Because no N entered the systems in hay or fertilizer during the study period, the main N inputs came with precipitation. Precipitation variation was greater in the 7-yr period than during the seven years before the study period (Table 1). This

resulted from an extremely high year (808, 626, and 1434 mm for the 1990–1991 growing, dormant, and annual periods, respectively) followed by an extremely low year (276, 415, and 691 mm for the 1991–1992 growing, dormant, and annual periods, respectively). The 7-yr annual average for mineral N in precipitation was approximately 9 kg ha⁻¹. This was slightly higher than the input measured during the seven years before the current study period, but the averages were well within a standard deviation unit. Although there was considerable variation in concentrations and inputs, as indicated by the large standard deviations, there did not appear to be any concentration or input trends over the 14 years of study.

Surface Runoff

Surface runoff was low (Table 2), even though there were slopes ranging up to 25% in these watersheds.

Table 2. Average seasonal and annual N concentrations (flow-weighted) and transport in surface runoff for the study period and the seven years before the study period.

		N concentration				N transport			
Time	Surface flow	NO ₃ -N	Mineral N†	Organic N	Total N	NO ₃ -N	Mineral N†	Organic N	Total N
	mm	mg L ⁻¹				kg ha ⁻¹			
Seven years before the study period (1983–1990)									
Pre-hay (Watersheds [WS] 104 and 129)									
Growing season‡	11 ± 16§	2.1	3.0	15.8	19.2	0.2	0.3	1.8	2.2 ± 4.4
Dormant season¶	11 ± 17	1.2	5.4	11.9	17.4	0.1	0.6	1.2	1.8 ± 4.5
Year#	22 ± 27	1.6	4.0	3.4	17.7	0.3	0.9	3.0	4.0 ± 7.2
Pre-grazing (WS 102 and 135)									
Growing season	3 ± 6	2.2	3.3	4.5	8.3	0.1	0.1	0.1	0.3 ± 0.4
Dormant season	4 ± 7	0.9	2.1	2.3	4.2	<0.1	0.1	0.1	0.1 ± 0.3
Year	7 ± 9	1.5	2.6	3.3	6.0	0.1	0.2	0.2	0.4 ± 0.5
Seven-year study period (1990–1997)									
Hay (WS 104 and 129)									
Growing season	2 ± 2	0.2	0.4	1.6	2.2	<0.1	<0.1	<0.1	<0.1 ± 0.1
Dormant season	6 ± 2	0.4	0.5	1.6	2.6	<0.1	<0.1	0.1	0.2 ± 0.2
Year	8 ± 8	0.3	0.5	1.7	2.5	<0.1	<0.1	0.1	0.2 ± 0.2
Grazing (WS 102 and 135)									
Growing season	3 ± 5	0.4	0.7	1.9	3.0	<0.1	<0.1	<0.1	<0.1 ± 0.1
Dormant season	5 ± 5	0.4	0.6	1.3	2.3	<0.1	<0.1	0.1	0.2 ± 0.2
Year	8 ± 8	0.4	0.6	1.6	2.6	<0.1	<0.1	0.1	0.2 ± 0.2

† Sum of NH₄-N, NO₃-N, and NO₂-N.

‡ May–October.

§ Values are means and standard deviations.

¶ November–April.

May–April.

With the exception of WS 129 in the pre-hay period, annual runoff averaged less than 10 mm. Before and during the pre-hay period (through April 1986), WS 129 had winter feeding of cattle as well as rotational summer grazing. This caused greater runoff and N transport for that period. As indicated by the standard deviations, there was considerable variability of N transport but average annual N transport in surface runoff during the study period was quite small, 0.2 kg ha⁻¹ or less.

Subsurface Flow

Seasonal and annual subsurface flows varied considerably through the pre-study and study periods, especially during the study period, as indicated by the large standard deviations (Table 3). The variation during the study period was greatly increased by the very high flow and very low flow during the first two years (Fig. 2). The average flow for the last five years was quite similar to the 7-yr average but with much less variation. The averages of the seasonal and annual flows did not differ greatly among the watersheds or between the two study periods. Therefore, differences in subsurface flow amounts among the watersheds during the study period would not be a major factor in causing differences in subsurface N concentrations and transport among the watersheds or between treatments.

With the pasture management practices used before the current study period, flow-weighted NO₃-N concentrations had exceeded 10 mg L⁻¹ in the subsurface flow from each of the watersheds (Fig. 3). With the cessation of N fertilizer applications in the three former summer grazing watersheds (WS 102, 104, and 135) and hay addi-

tions in WS 129, NO₃-N concentrations in the ground water began to decrease. Following five years of no N fertilizer applications, the NO₃-N concentrations in the ground water were as low as they were when the annual N fertilizer applications went from 56 to 168 kg ha⁻¹, which was below 5 mg L⁻¹. The exception was WS 129. Because of the additions of N from hay, WS 129 already had NO₃-N levels approaching 10 mg L⁻¹, and it was necessary to go farther back in its management history to have these low levels (Owens et al., 1982).

Seasonal subsurface NO₃-N concentrations of WS 102, 104, and 135 visually increased from 1982 to 1990 (Fig. 3). The trends of increasing concentrations from regression analysis during the pre-study period ranged from 0.75 mg L⁻¹ season⁻¹ at WS 102 to 1.07 mg L⁻¹ season⁻¹ for the three summer grazing watersheds and 1.74 mg L⁻¹ season⁻¹ at WS 129 (Table 4). Coefficients of determination for this period were large (ranging from 0.73 to 0.87; Table 4), and all regressions were statistically significant at less than the 0.0001 level. Seasonal concentrations decreased significantly after 1990 during the study period at the three sites (Fig. 2 and Table 4). Trends of decreasing concentration ranged from -2.24 mg L⁻¹ season⁻¹ at WS 129 to -0.71 mg L⁻¹ season⁻¹ at WS 102. All decreasing trends for the watersheds during the study period were statistically significant with regression probabilities at less than 0.0004 and coefficients of determination ranging from 0.66 to 0.94. For WS 104, 129, and 135, seasonal concentration decreased about 30% faster than it increased during the pre-study period, but was about the same for WS 102 for the two periods.

Table 3. Average seasonal and annual subsurface flow for the study period and the seven years before the study period.

Time	Subsurface flow			
	Grazing		Hay	
	Watershed 102	Watershed 135	Watershed 104	Watershed 129
	mm			
	Seven years before study period (1983–1990)			
Growing season†	54 ± 29‡	60 ± 43	57 ± 33	93 ± 39
Dormant season§	233 ± 71	217 ± 56	232 ± 72	236 ± 54
Year¶	287 ± 75	277 ± 68	289 ± 78	329 ± 52
	Seven-year study period (1990–1997)			
	1990–1991			
Growing season	189	197	195	246
Dormant season	268	427	273	396
Year	457	624	468	642
	1991–1992			
Growing season	36	36	35	19
Dormant season	10	10	10	9
Year	46	46	45	28
	1992–1997			
Growing season	78 ± 43	76 ± 42	74 ± 44	75 ± 42
Dormant season	216 ± 63	219 ± 68	209 ± 55	222 ± 65
Year	294 ± 88	295 ± 93	283 ± 87	297 ± 89
	1990–1997			
Growing season	88 ± 57	87 ± 59	85 ± 59	92 ± 75
Dormant season	194 ± 94	219 ± 126	190 ± 89	216 ± 117
Year	282 ± 134	306 ± 174	275 ± 135	308 ± 181

† May–October.

‡ Values are means and standard deviations.

§ November–April.

¶ May–April.

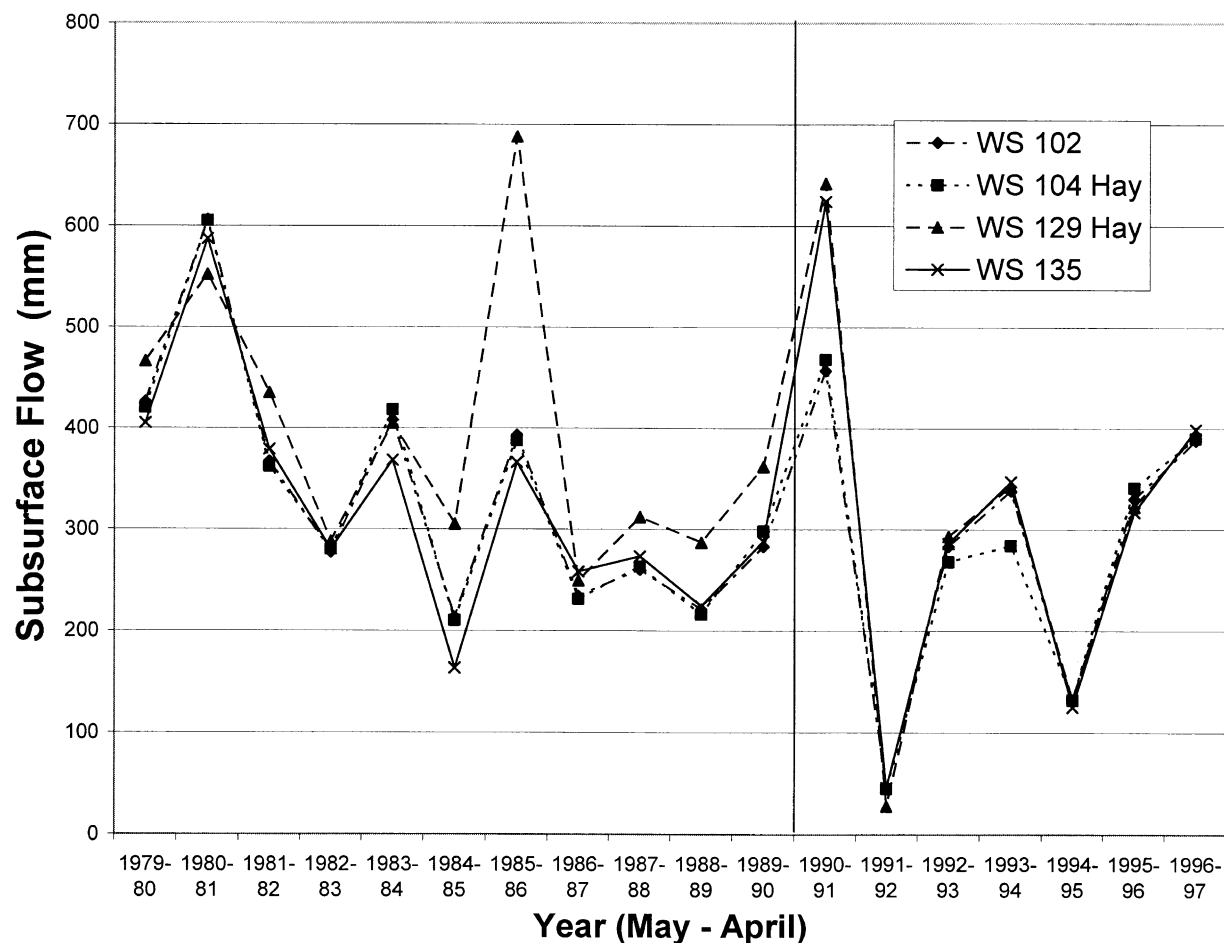


Fig. 2. Annual subsurface flow throughout the 11-yr pre-study period (1979–1990) and the 7-yr study period (1990–1997).

These are small aquifers with shallow soils, which would have less potential for N storage than deeper soils. Therefore, N concentration changes can be observed rather quickly (i.e., in only a few years). In larger aquifers, especially with deeper soils, such concentration changes would be expected to occur much more slowly. The Coshocton lysimeters represent even smaller “systems,” even though they are large (surface area of 8.1 m² and a depth of 2.4 m) compared with most lysimeters. Nitrate N concentration trends were similar to those observed on the small watersheds but occurred more quickly (Owens et al., 1999). The increase in N concentration in the lysimeter percolate and the decrease resulted from high N fertilizer rates and omission of N fertilizer, respectively.

Nitrate N concentrations in the shallow ground water began to decrease while N was still being applied before the current study period (Fig. 3). A closer examination of the data showed that instead of an early concentration decrease occurring, the N concentrations for the 1989 dormant and 1990 growing seasons were unusually high. This resulted from a dry summer in 1988. The subsurface flows during the 1988 growing season were the lowest (except for WS 129) of all the growing seasons from 1974 through 1996. Each of these watersheds were in forage systems all of those years. During that dry sum-

mer, less NO₃ leaching occurred and probably less plant uptake of N. Thus, much more than the usual amount of N was available for leaching when normal precipitation and subsurface flow resumed. Tyson et al. (1997) noted that the highest N leaching losses from fertilized grass pastures occurred following dry summers.

Although two different management techniques were used to study the rate of lowering NO₃-N concentration in ground water, haying and grazing, three of the watersheds reached a NO₃-N level below 5 mg L⁻¹ almost simultaneously. Nitrate N concentrations in the subsurface flow from WS 102 were consistently lower than the other watersheds and reached this level sooner. Even though watershed characteristics and pre-study treatments had WS 102 and 104 and WS 129 and 135 as “matched pairs” in regard to subsurface flow (Table 3) and N transport, NO₃-N concentrations still reached similar levels after five years of no N fertilizer applications. The high levels of N brought into WS 129 with hay produced a higher rate of increase of NO₃-N concentration in subsurface flow than in the other three watersheds, but it also had the highest rate of NO₃-N concentration decrease with no N fertilizer inputs (Table 4). Although there were high NO₃-N levels in the ground water from different management practices on these watersheds, this study showed that either haying

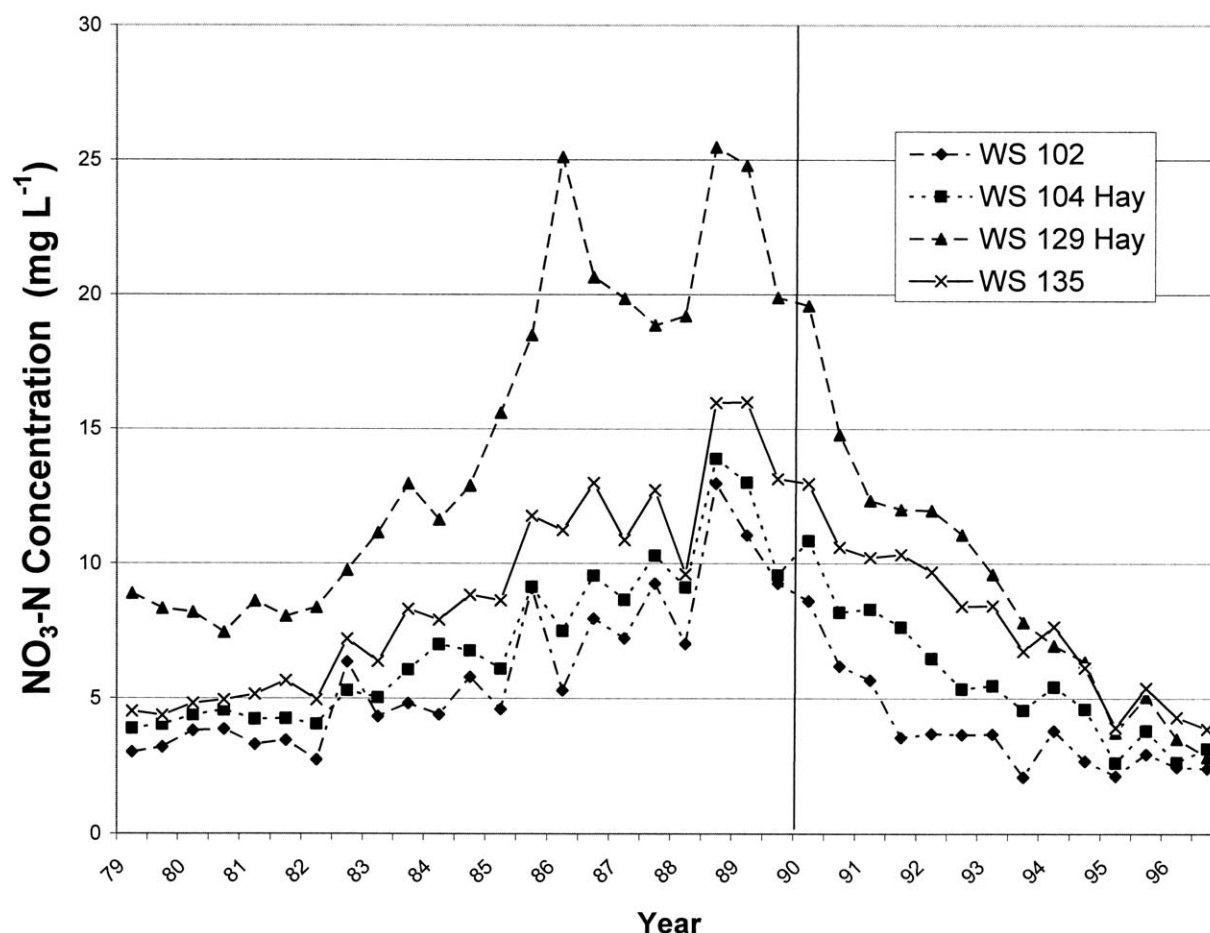


Fig. 3. Average flow-weighted seasonal (growing and dormant) concentration of NO₃-N for subsurface flow throughout the 11-yr pre-study period (1979–1990) during which 168 kg N ha⁻¹ was applied annually to rotationally grazed pastures, and the 7-yr study period (1990–1997) during which no N fertilizer was applied and two watersheds were grazed and two were hayed. Each year notation is a grazing year (May–April).

or grazing without N fertilizer inputs will meet the need of reducing NO₃-N concentrations in ground water, even from heavily N loaded areas.

As NO₃-N concentrations increased with fertilizer applications and decreased with no N fertilizer, NO₃-N transport also increased and decreased, respectively (Fig. 4). Although there were extreme years for subsurface flow, there was no subsurface flow trend during the study and pre-study periods (Fig. 2). But the increasing trend in annual NO₃-N concentrations produced an increasing trend in annual NO₃-N transport during the pre-study period. The peak year of NO₃-N transport (1990–1991) followed by a very low year of NO₃-N transport (1991–1992) largely resulted from those two years being extreme hydrologic years (Table 3, Fig. 2). The high subsurface

flow in the 1990–1991 year had more of a flushing effect than would have occurred in a more nearly average flow year. However, a detailed examination of the effects of extreme hydrologic events is beyond the intended scope of this article.

Other Nitrogen Loss Pathways

Initially we expected the N in the ground water to decrease more quickly in the hayed watersheds because of the N removed from the system with the hay removal. An annual average of 59 and 58 kg N ha⁻¹ was removed in hay from the two hayed watersheds (WS 104 and 129, respectively). When N fertilizer applications were stopped, the amount of hay produced, and N removed,

Table 4. Rates of increase and decrease of NO₃-N concentrations, coefficients of determination, and significance probabilities for the four watersheds.

Watershed	Seven years before study period (1983–1990)			Seven-year study period (1990–1997)		
	Rate of increase†	r ²	Probability	Rate of increase†	r ²	Probability
	mg L ⁻¹			mg L ⁻¹		
102	0.75	0.73	<0.0001	-0.71	0.66	0.0004
104	0.83	0.83	<0.0001	-1.09	0.89	<0.0001
129	1.74	0.83	<0.0001	-2.24	0.93	<0.0001
135	1.07	0.87	<0.0001	-1.31	0.94	<0.0001

† Rate of increase or decrease per season.

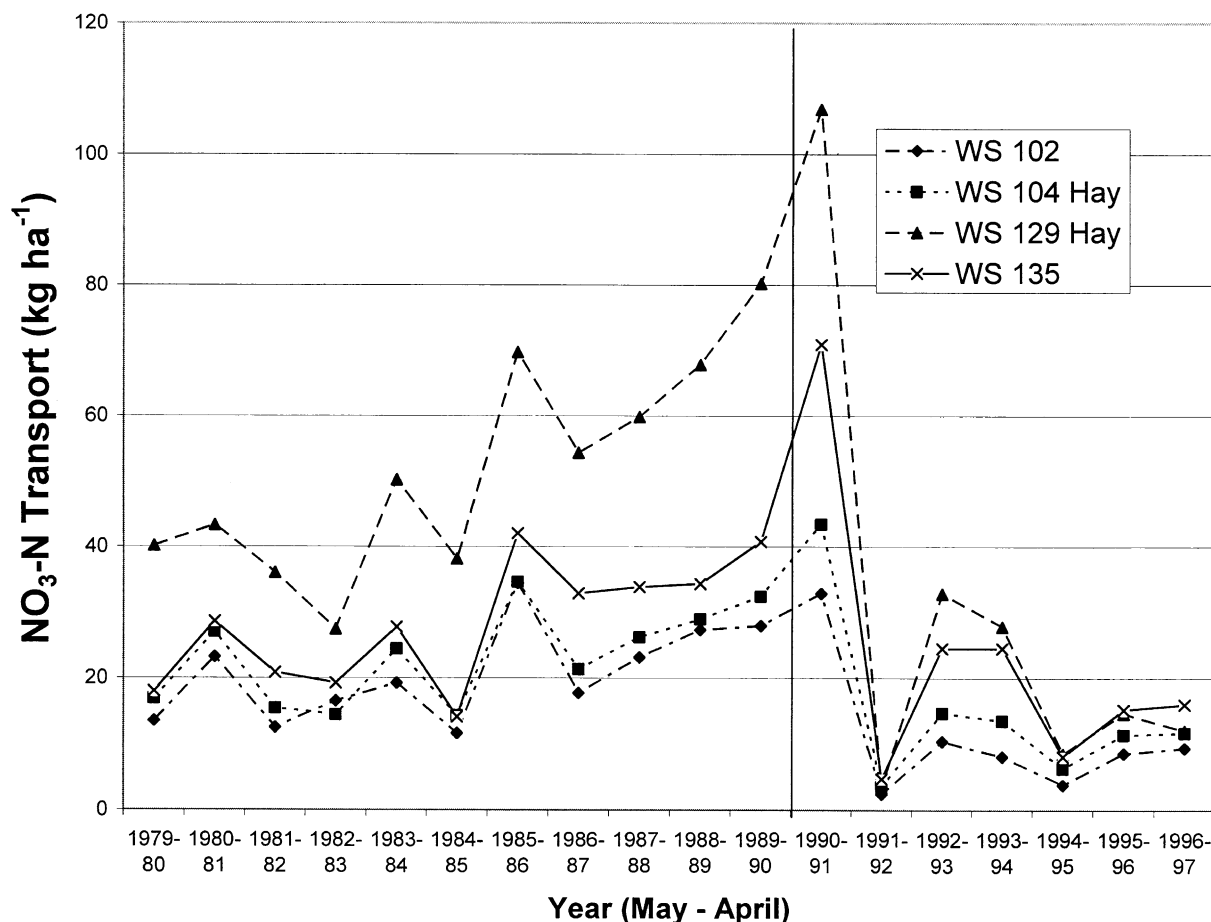


Fig. 4. Annual NO₃-N transport in subsurface flow throughout the 11-yr pre-study period (1979–1990) during which 168 kg N ha⁻¹ was applied annually to rotationally grazed pastures, and the 7-yr study period (1990–1997) during which no N fertilizer was applied and two watersheds were grazed and two were hayed. Each year notation is a grazing year (May–April).

gradually decreased during the study period (Table 5). The decline in loss of N via hay removal coincides with lower N concentrations in ground water because of less N available to leach.

With the ground water N concentrations declining similarly under the grazing watersheds and the hayed watersheds during the study period, similar losses of N from the systems would be expected. The “crop output” from the grazed areas was the calf crop from the cow-calf herd. Using weaning weights and 19.2 g kg⁻¹ for the N composition of the live calves (Odongo et al.,

1984), the annual N removal via calves was calculated to be 4.2 kg ha⁻¹ during the study period.

Gaseous N losses through N₂O emissions, NH₃ volatilization, and denitrification increase with N inputs (Ryden, 1986; Sommer and Hutchings, 1997; Whitehead, 1995, p. 163–164). Significant N losses can occur from urine and feces deposits with losses from urine being much greater than from feces (Ryden, 1986; Whitehead, 1995, p. 163). In this study, N losses via each of these pathways probably decreased as the N in the systems decreased during the course of the study. Nevertheless, gaseous N losses combined with “calf crop” removal could account for N losses approaching the N losses from the hayed watersheds.

Nitrate N levels in ground water exceeded 10 mg L⁻¹ in all four watersheds because of high N inputs. Watershed 129 had N inputs both from mineral fertilizer and hay. Thus it was somewhat different from the other three watersheds, but all the watersheds had NO₃-N levels above the maximum contaminant level. Both haying and grazing with no N inputs reduced N leaching and lowered NO₃-N concentrations in the ground water. In a grazing period before this 14-yr period of study (Owens et al., 1982), 56 kg N ha⁻¹ produced low NO₃-N levels in ground water, 3 to 4 mg L⁻¹. This level of N input could be used during a period to lower NO₃ leach-

Table 5. Forage dry matter, N content, and amount of N removed from the hay watersheds, 1990–1996.

Year	N†	Hay, Watershed 104		Hay, Watershed 129	
		DM‡	N	DM	N
	g kg ⁻¹	kg ha ⁻¹			
1990	15.3	5447	83	4466	68
1991	17.6	4834	85	5617	99
1992	20.1	2788	56	2535	51
1993	11.2	3882	44	4572	51
1994	16.9	2915	49	3115	53
1995	14.7	2765	41	2359	35
1996	17.6	3127	55	2947	52
Average N			59		58

† Forage samples were not separated by watershed for analysis.

‡ Dry matter.

ing, although the rate of reduction would probably not be as fast as with no N inputs. Using the collective research at the North Appalachian Experimental Watershed (Owens et al., 1982, 1983, 1992, 1994), we suggest that N inputs for grazing systems should not exceed 100 kg N ha⁻¹ annually to maintain ground water NO₃-N concentrations below 10 mg L⁻¹. All sources of N (e.g., fertilizer, hay, supplemental feed, legumes, and manure) should be included in the total. A higher N rate could probably be used in a haying system without excessive leaching. This is currently being evaluated at the North Appalachian Experimental Watershed. Although an annual rate of N input of 100 kg ha⁻¹ should be an environmentally friendly rate, this would be too high a rate to be very effective in reducing N leaching or lowering high NO₃-N levels in ground water.

CONCLUSIONS

Nitrate N concentrations in excess of 10 mg L⁻¹ can be found in ground water under pastures that receive high N inputs. Two management options, rotational grazing and hay removal, with no N inputs will allow NO₃-N concentrations in ground water to be reduced. Peak ground water NO₃-N concentrations of 13 to 17 mg L⁻¹ went to less than 5 mg L⁻¹ within five years of omission of N fertilizer. The ground water NO₃-N concentration from a more heavily N loaded small watershed decreased from 25 to less than 5 mg L⁻¹ in six years. Nitrate N transport also decreased as NO₃-N concentrations decreased. For these two practices to be effective, there should be little or no N inputs so that the N in the system can be reduced. Thus, N available for leaching will be reduced. When ground water NO₃-N concentrations are above maximum contaminant levels, a livestock producer can achieve lower NO₃ losses and acceptable ground water NO₃-N quality under haying or continued grazing with low or no N inputs, even in an area with previous high N loading.

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